1 Nutrients and eutrophication in lakes

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1.1 Introduction

Within Europe, eutrophication has had a high profile since the early 1980s when wide-spread occurrence of blue-green algal blooms in standing and slow-flowing freshwaters gave rise to considerable interest and concern by the public, the media and by water industries. Of all surface waters, the impact of nutrient enrichment on lakes and reservoirs has probably received the greatest attention and is well understood in general qualitative terms. Nutrients tend to be a secondary and indirect driver of community composition but are a significant driver of productivity, leading to increased growths of algae (phytoplankton and filamentous algae) and aquatic plants. The increase in phytoplankton crops can lead to secondary problems, such as the loss of sensitive macrophyte and fish species. These impacts, as well as the increasing frequency and intensity of toxic cyanobacteria blooms, have become a widespread problem in European lakes and have considerable consequent environmental, social and economic implications.

These general effects of increased nutrient loading, or eutrophication, have been incorporated within the definition of eutrophication in recent European legislation - Urban Waste Water Treatment Directive (UWWTD) and Nitrates Directive (ND) and agreements (Oslo - Paris Marine Convention, OSPAR). These define eutrophication as:

'The enrichment of water by nutrients, especially compounds of phosphorus (P) and/or nitrogen (N), causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned'

One problem with this definition of eutrophication is that it cannot easily be quantified. Eutrophication is a process, not a state such as 'eutrophic', the latter being more readily defined in terms of phytoplankton biomass (e.g. as a concentration of chlorophyll-*a*) or a concentration of a nutrient such as P. In the WFD, this issue of how to assess the degree of change in freshwater quality is tackled by relating (ecological) quality to a baseline or reference state under minimal human influence. The requirement within the WFD to define nutrient conditions for high and good ecological status requires a specific quantitative dose-response understanding of how nutrient conditions relate to biological quality in individual lake ecotypes.

The impact of nutrient pressures in a particular lake ecotype or individual site can be dependent on a number of 'sensitivity' factors. Deep lakes are often more predictable in terms of nutrient effects on phytoplankton abundance and the indirect impacts on profundal (deep-water) invertebrate and fish communities. In shallow lakes, ecological thresholds in response to nutrient conditions are much more difficult to define due to the feedback effects of macrophytes and fish on lake community structure. In addition, nutrient enrichment is frequently accompanied by other pressures and it is therefore often difficult to identify ecological impacts solely due to nutrient conditions, let alone impacts due to particular nutrients (N or P).

This chapter considers both the known qualitative and quantitative relationships between nutrient conditions and direct or indirect biological responses, in lake ecosystems. It considers effects on phytoplankton, phytobenthos, macrophytes, benthic invertebrates and fish. The effects of nutrients acting in combination with other pressures (hydromorphological, acidification and toxic) are considered in chapters 5, 6 and 12.

1.2 Phytoplankton

Introduction

The phytoplankton community is widely considered the first biological community to respond to eutrophication pressures and is the most direct indicator of all the Biological Quality Elements (BQEs) of nutrient concentrations in the water column. There are numerous socio-economic problems associated with increases in phytoplankton abundance, particularly with increasing frequencies and intensities of toxic cyanobacteria blooms. These include detrimental effects on drinking water quality, filtration costs for water supply (industrial and domestic), water-based activities and conservation status (e.g. sensitive pelagic fish species, such as coregonids). In some contexts, however, increasing phytoplankton abundance can be considered as a positive feature, for example, in increasing fisheries productivity, and some phytoplankton taxa can be considered indicative of high ecological status.

Annex V of the WFD outlines three features of the phytoplankton quality element that need to be considered in the assessment of the ecological status of lakes and for which there is thus the need to relate in quantitative terms to nutrient conditions, i.e. for which dose-response curves need to be investigated. These three are:

- Phytoplankton composition
- Phytoplankton abundance and its effect on transparency conditions
- Planktonic bloom frequency and intensity.

The WFD normative definition indicates that declining ecological quality is associated with increasing phytoplankton abundance, greater proportions of cyanobacteria and more frequent and intense phytoplankton blooms. These three features are considered further in the context of their specific use as metrics of nutrient pressures.

Phytoplankton composition

Individual species or taxa can be considered as positive, negative or indifferent indicators in relation to nutrient pressures. Negative indicators include species of green algae (e.g. *Scenedesmus*) and diatoms (e.g. *Stephanodiscus*) and many groups of cyanobacteria, such as the large colonial and filamentous genera *Microcystis, Aphanizomenon* and *Anabaena*. The latter are favoured by relatively stable stratification and high alkalinity and can, therefore, form a significant natural component of the phytoplankton community in deep alkaline lakes. As nutrient concentrations increase, however, the dominance and abundance of cyanobacteria, in particular, generally increases often resulting in dense, mono-specific blooms during summer (Reynolds, 1984). The proportion of cyanobacteria of the total phytoplankton biomass is a metric that is being recommended in Norway with regard to WFD requirements and for which thresholds can be generated (Figure 1.1; Lyche Solheim *et al.*, 2004). Positive indicators include species of chrysophytes (e.g. *Dinobryon*), desmids (e.g. *Cosmarium*) and some diatoms (e.g. *Cyclotella comensis*).

The ratio of positive to negative species can be used as a metric of ecological status and some Member States (Denmark - Sondergaard *et al.*, 2003; UK – Carvalho *et al.*, 2004) are developing metrics based on the relative abundance of positive and negative phytoplankton functional groups (as outlined in Reynolds *et al.*, 2002), rather than taxonomic units. Both taxonomic (Figure 1.1) and functional group (Figure 1.2) dose-response relationships need to be developed and validated further.

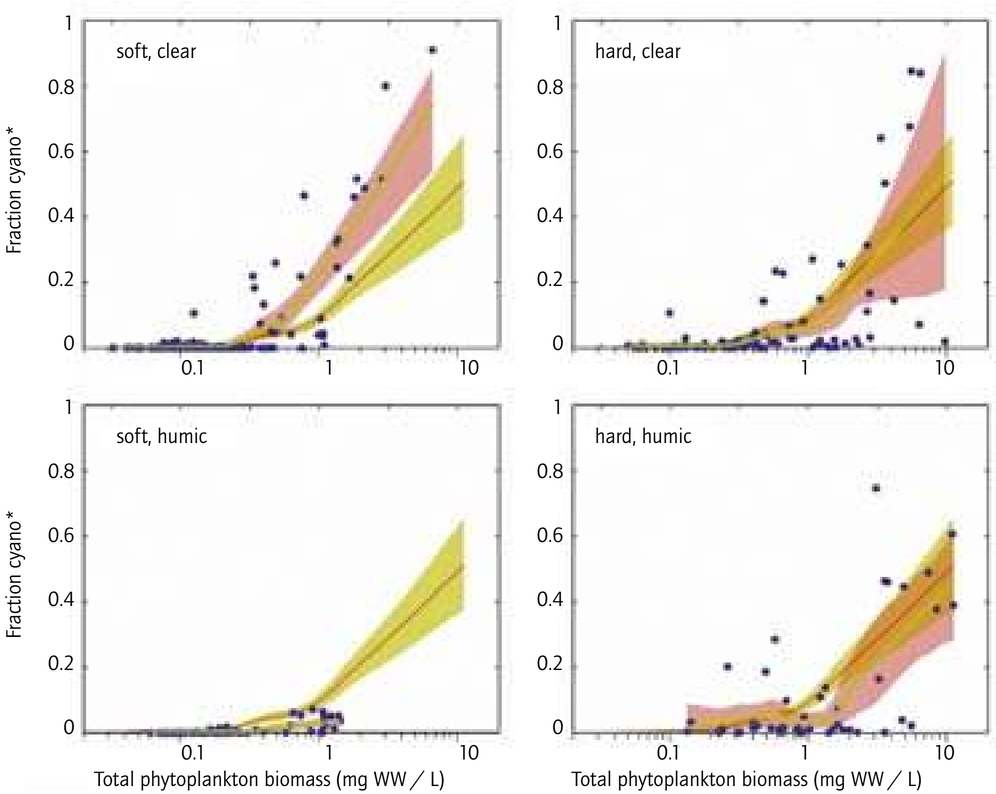

Figure 1.1 Proportion of impact phytoplankton indicator Cyanobacteria as a function of total phytoplankton biomass is shown for four different Norwegian lake types. Yellow lines and pink confidence bands are LOWESS-based trendlines for the single water type, while the trendline for the complete dataset (not divided into lake type)s are shown in all graphs as red lines and yellow confidence bands (Lyche Solheim *et al.*, 2004).

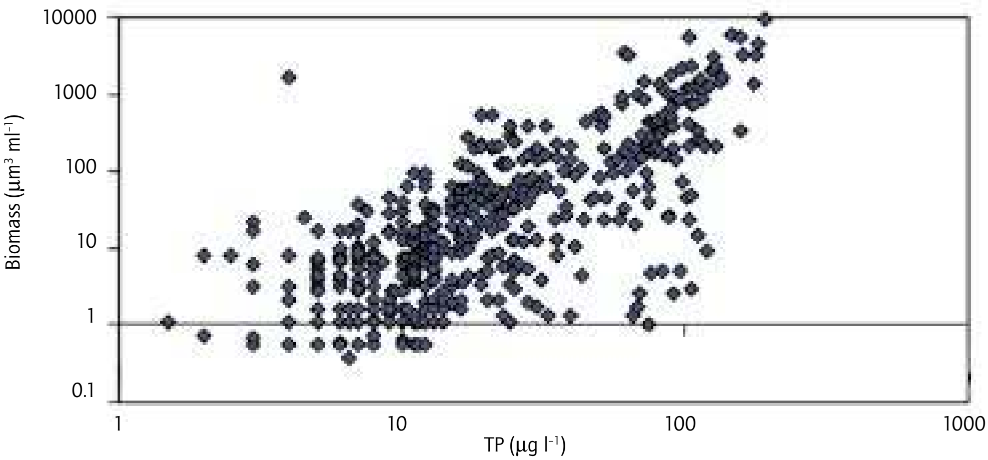

Figure 1.2 Relationship between biomass of the 'impact' phytoplankton functional group "Lm" (c.f. Reynolds et al., 2002) and total phosphorus concentrations in Finnish lakes (courtesy by Lepistö *et al.*, unpublished).

Phytoplankton abundance and its effect on transparency conditions

In general, as nutrient concentrations increase, phytoplankton abundance shows more frequent and sustained peaks throughout summer, and consequently transparency declines. The abundance of phytoplankton can be expressed in three ways:

- Density of individual organisms (numbers of individual cells/filaments/colonies per ml)
- Volume of these cells as a fraction of the volume of water (biovolume in mm^3 m^{-3})
- Chlorophyll-*a* concentration expressed as a concentration in water (μg l^{-1}).

All of these approaches have their pros and cons. For WFD monitoring purposes, many Member States are choosing to use chlorophyll-*a* concentration, as it is a relatively robust and simple measure widely used in all Member States (Sondergaard *et al.*, 2003; Carvalho *et al.*, 2004).

Transparency is widely used as an indirect or surrogate estimate of the amount of phytoplankton or chlorophyll-*a*, and, therefore, as an indicator of eutrophication. The optical properties of lakes are, however, not just controlled by the amount of phytoplankton, other factors such as dissolved colour, suspended inorganic and organic particles may contribute significantly in some lakes (Tilzer, 1988).

Transparency is mentioned in Annex V of the WFD as a general physico-chemical factor supporting the biological elements. In the normative definition of ecological status (Annex V, 1.2.2) it is stated that, under high status conditions, *"the average phytoplankton biomass is consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions"*.

The transparency of water is commonly estimated using a Secchi disk, less commonly it is measured using remote sensing methods. The Secchi disk measurement is obtained for a very restricted area (a lake or more specifically discrete points in a lake) while remote sensing methods can cover a larger area at once (up to hundreds of lakes). A Secchi disk is a circular, 20 cm diameter, black and white disk. The Secchi depth is the average of the depths where the disk disappears when lowered into water and reappears again when raised. The actual photosynthetic layer of water is approximately three times the Secchi disk depth. The Secchi disk method is a very simple, useful and cost-effective way to monitor and assess the status of surface waters. Furthermore, this method is a useful tool for monitoring by citizens to enhance public participation in the WFD. The use of satellite remote sensing is, however, a cost-effective method to assess the transparency of surface waters (turbidity, chlorophyll-*a*) on a large scale.

Quantitative relationships have been developed relating total phosphorus (TP) concentrations with phytoplankton biomass (chlorophyll-*a*) and water clarity (Secchi depth). The most widely reported relationships are those developed by Dillon and Rigler (1974) and Vollenweider OECD (OECD, 1982). The latter relationship (Figure 1.3) was developed for a set of, predominantly large, temperate lakes, not ecotype specific. There is a great deal of scatter in all the published relationships highlighting the fact that a number of sensitivity factors are involved, such as water colour and flushing rate, of which the latter can be altered by hydromorphological pressures such as flow regulation. There is, therefore, a clear need to develop more complex regression models that incorporate these sensitivity factors. The importance of such sensitivity factors highlights the need to incorporate lake typology factors (e.g. depth, altitude, colour) into models to understand how nutrients are transformed into algal biomass, highlighting the value of lake-type specific models as advocated by the WFD.

Planktonic bloom frequency and intensity

The term “planktonic bloom” refers to the phenomenon where phytoplankton populations greatly increase in number over a limited period of time. Most concern focuses on periods when these blooms are composed of toxic cyanobacteria that can form surface scums and accumulate on lake

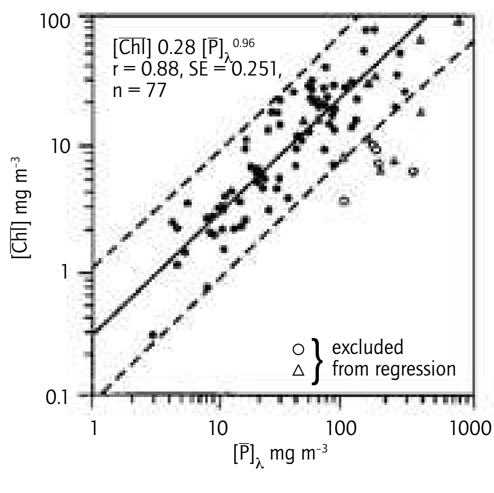

Figure 1.3 Regression of annual mean phosphorus concentration with phytoplankton biomass (as annual mean chlorophyll-*a*; from OECD, 1982).

shores. The development of dense "bloom" populations of cyanobacteria is more predictable in deep lakes, compared with shallow lakes, as bloom frequency and intensity is particularly affected by physical factors such as stratification intensity and the depth of light penetration, however, no quantified relationships have been described in the literature detailing how bloom frequency or intensity is related to nutrient conditions. Furthermore, there is no agreed European quantitative limit for defining simply when an algal bloom is present and different Member States use a variety of thresholds based on measures of cyanobacteria abundance, chlorophyll-*a* concentrations or amounts of toxins (e.g. microcystin). A more widely accepted definition of what constitutes a phytoplankton bloom is needed before dose-response relationships can be developed.

Classification schemes

The most widely used classification scheme is that developed by the OECD (1982), which is summarised in Table 1.1. Many other regression models or classification schemes have been published which incorporate relationships between nutrients (exclusively phosphorus) and phytoplankton (usually abundance) and transparency, summarised in Table 1.2.

Table 1.1 Lake classification scheme based on trophic status (OECD, 1982).

	Annual mean Total P ($\mu g.l^{-1}$)	Annual mean Chlorophyll-*a* ($\mu g.l^{-1}$)	Annual maximum Chlorophyll-*a* ($\mu g.l^{-1}$)	Annual mean Secchi depth (m)
Ultra-oligotrophic	<4	<1	<2.5	>12
Oligotrophic	<10	<2.5	<8	>6
Mesotrophic	10-35	2.5-8	8-25	6-3
Eutrophic	35-100	8-25	25-75	3-1.5
Hyper-eutrophic	>100	>25	>75	<1.5

Models

Models simulating the impact of nutrients on the phytoplankton community can be divided into three main categories:

1. Empirical statistical models
2. Mechanistic/deterministic models
3. Theoretical food-web models.

Many empirical statistical models have been developed based on correlations and regressions of observed data on nutrient and chlorophyll-*a* concentrations and Secchi disc depth. Only one published model relates Secchi depth to phytoplankton biomass (Hillbricht-Illkowska, 1993). Canfield and Hodgson (1983), besides chlorophyll and Secchi disk depth, included dissolved organic matter concentration to account for water colour in their model.

Table 1.2 Published phytoplankton models or classification schemes for lakes.

Reference	Region	Notes
Dillon and Rigler (1974)	North America	Regression based on mean summer chlorophyll and spring TP concentrations from 19 lakes in Canada combined with published data from other North American lakes. Regression used to predict summer chlorophyll from a single spring measurement of TP.
OECD (1982)	Several	Lake classification scheme based on expert evaluation of lake data. Classification defines boundary values of total phosphorus, chlorophyll and transparency (Secchi depth) between trophic classes from ultra-oligotrophic to hyper-eutrophic. The classification is based on data from a large number of lake types from regions across the globe, although there was a predominance of large, deep lakes in the original dataset.
Gibson *et al.* (2000) based on Carlson (1977; 1980)	USA	EPA lake classification scheme which uses phytoplankton abundance, chlorophyll concentrations and transparency (Secchi disc depth).
Premazzi and Chiaudani (1992)	Europe	Classification scheme based on annual mean and maximum chlorophyll concentrations defined for complying (excellent and good status) and non-complying waters (fair, poor, bad status).
Cardoso *et al.* (2001)	Europe	Threshold chlorophyll *a* concentrations for lakes subject to eutrophication set in response to the UWWTD.
Moss *et al.* (1996) Heinonen (2000) Bratli (2000) Swedish EPA (2000) Newman (1988)	Great Britain Finland Norway Sweden Denmark	Transparency (Secchi depth) used as a parameter in classification of water quality in many European countries.
Bennion (1994) Wunsam and Schmidt (1995)	UK European Alps	Quantitative direct relationships between phytoplankton composition and nutrient conditions for planktonic diatoms, based on relative abundances within surface sediment sub-fossil assemblages. Individual species TP optima are dataset specific and cannot necessarily be applied outside the region of development or for different lake types.
Moss *et al.* (2003) Sondergaard *et al.* (2003) Carvalho *et al.* (2004) Lyche Solheim *et al.* (2004)	Europe Denmark UK Norway	Preliminary phytoplankton classifications (composition and abundance) developed specifically for the WFD.

The nutrient used in the published regressions is almost exclusively in-lake total phosphorus concentration and the regressions are usually based on logarithmic transformations (Carlson, 1977; OECD, 1982; Mazumder and Haves, 1998; Portielje and van der Molen, 1999), only one is based on a power function (Sas, 1989). In-lake TP concentrations can also be estimated using data on lake depth, area, flushing rate and external TP loads using equations of Dillon and Rigler (1974) and Kirchner and Dillon (1975). By linking the two, it is therefore possible to model how changes in flushing rate and catchment land-use could affect phytoplankton abundance. Deep lakes tend to show a much more predictable response than shallow lakes; it is therefore possible that a relatively precise empirical model could be derived for deep European lakes to predict impacts of changing nutrient conditions.

Mechanistic models which explore relationships between the physico-chemical environment and the phytoplankton community are potentially useful for identifying reference conditions or evaluating the effectiveness of restoration measures. A number of mechanistic models have been developed which predict phytoplankton abundance and composition (DBS Model: PC Lake: Janse and Van Liere, 1995; Los *et al.*, 1997; Reynolds, 1999; PROTECH model: Elliott *et al.*, 2001). Many of these models incorporate mixing regimes and are capable of simulating multiple resource limitation (phosphorus, nitrogen, etc.) and grazer effects.

Dynamic, deterministic models have also been developed to determine light attenuation in the water column associated with phytoplankton (Kirk, 1994). Models such as CAEDYM, DYRESM, SOBEK calculate light attenuation in the water column using Beer's law with varying light extinction coefficients for different algal groups, suspended sediments and dissolved organic matter. From light attenuation, many models further calculate transparency. Theoretical models examining lake ecosystem structure and functioning, in particular issues of stability and resistance in shallow lakes, have been developed by Scheffer (1998).

1.3 Macrophytes

Introduction

The macrophyte community is generally regarded as a key indicator of the ecological status of lakes as macrophytes provide habitat for many other aquatic biota (e.g. fish, macro-invertebrates, wetland birds) to feed, seek refuge, or breed. Macrophytes are relatively long-lived organisms (months to years), compared with phytoplankton and invertebrates. Macrophytes have a very limited motility and are intrinsically linked to the prevailing environmental conditions in both the surrounding lake water and sediments, through their roots and leaves. Individual species are sensitive to physical and chemical changes in these media and hence are good indicators of both current environmental conditions and longer-term environmental changes. Macrophytes, in terms of assessing eutrophication pressure, can indicate enhanced nutrient concentrations through the direct effects on species growth (biomass) and through indirect effects on species composition, as may be caused by a reduction in transparency associated with nutrient-related increases in phytoplankton and epiphyton.

Transparency is a key physico-chemical factor controlling the distribution and abundance of submerged macrophytes in lakes. Changes occurring in the intensity and quality of light as it passes down through the aquatic vegetation itself may exert a crucial effect on the development and structure of aquatic macrophyte communities, as well as on the photosynthetic efficiency and productivity of the vegetation (Sculthorpe, 1967). Aquatic macrophytes are profoundly influenced by the transmission of light (the photosynthetically active radiation (PAR) through a water body, which impacts upon photosynthetic processes. In turbid conditions the PAR passing through the water can be attenuated by the scattering and absorption of light by suspended particles, both organic and inorganic. Increasing turbidity can often result in a shift in community from submerged to floating leaved or emergent species (Hough and Forwall, 1988). However, in terms of its impact on aquatic macrophytes, turbidity can be difficult to separate from other properties of a lake, e.g. colour and shading. Indeed, little is known of the precise influence of underwater changes in light intensity and quality on the distribution of life forms and species of aquatic macrophytes. It may be that some plants of a particular habitat may tolerate, or even prefer sustained low intensities and/or deficiencies in certain wavelengths. Phenotypical variations due to abiotic circumstances have been recorded for several species under changing inundation conditions e.g. Nabben *et al.* 1999; Van den Berg *et al.* 1999. Hyper-eutrophication may even lead to deterioration of emergent and floating leaved communities due to reducing conditions in the sediment. These non-linear responses of species composition and abundance to eutrophication pressures complicate the indicator value of aquatic macrophytes.

Annex V of the WFD outlines two macrophyte-related quality elements that need consideration in the assessment of the ecological status of lakes:

- macrophyte community composition
- macrophyte abundance.

Annex V of the WFD includes transparency as a physico-chemical element for supporting this assessment of macrophyte communities. The WFD normative definitions for high, good and moderate ecological status determine that ecological quality is considered as declining when the composition of macrophytic taxa differ moderately from the type-specific communities and are significantly more distorted than those observed at good quality (slight changes in composition and abundance of macrophytic taxa compared to the type-specific communities).

Metrics

The macrophyte community composition of lakes can be broadly related to a nutrient gradient, as individual species appear to show differing tolerances to nutrient conditions, although research suggests that other factors such as alkalinity, sediment type and water depth may be more important in determining species composition (Penning, pers comm.). Some species such as *Isoetes lacustris*, *Potamogeton polygonifolius* and *Sparganium angustifolium* tend to be restricted to low alkalinity, oligotrophic lakes, other species, for example, *Lemna* spp, *Myriophyllum spicatum* and *Potamogeton pectinatus* are typical of high alkalinity, (natural) eutrophic lakes. However, many macrophyte species can be regarded as being indifferent to a change in trophic level, occurring over a wider range of trophic conditions.

The relationship between the abundance of macrophytes and nutrients is complicated by internal interactions between other ecological components of the ecosystem, particularly in shallow lakes (Jeppesen *et al.*, 2000; Moss *et al.*, 2003). For example, Scheffer *et al.* (1993) produced an "alternative stable states" model to describe shallow lake functioning. Rather than exhibiting linear responses to changing nutrient concentrations, macrophyte species may undergo a threshold response whereby they abruptly change from being abundant to a state where they are absent, when phytoplankton dominate and vice versa. It appears that this can be due to initial spring conditions, but also the characteristics of the fish community and management thereof can play an important role (Van de Bund and Van Donk, 2002). Abundance can also be calculated as the coverage of aquatic vegetation as a percentage of potential littoral area (USEPA, 1998; Anonymous, 2003). Finnish case studies showed that helophytes and floating leaved vegetation coverage could be related to an increase of phosphorus and nitrogen of lake water (Kanninen *et al.*, 2003; Leka *et al.*, 2003). In shallow lakes macrophytes may even disappear when a hypereutrophic, algal dominated state is reached (Blindow, 1992).

Although no comprehensive study has been undertaken to relate aquatic macrophytes with turbidity/transparency measurements, a number of studies have examined the depth range of individual species (e.g. Spence and Chrystal, 1970; Blindow, 1992; Wilby *et al.*, 2000) or focused on specific locations such as the Danube Delta, where Coops *et al.* (1999) classified lakes into three different types that were related to turbidity ranges. In general, in relative terms, it appears that the deeper a species grows, the less light the species requires and, therefore, the assumption could be made that deeper growing species can compete better for light than other species and will be more resistant to increased turbidity. However, not all species tolerant to low light and growing at greater depths in clear lakes will successfully colonise lakes with high turbidity. Exceptions may be species that are sensitive to factors such as eutrophication, substrate type, siltation of leaves, or light quality, rather than just quantity of light (Davis and Brinson, 1980).

The sensitivity of North American submerged macrophytes to turbidity has been explored using the Turbidity Tolerance Index, i.e. the ratio of the depth maxima of a species to the Secchi disc transparency depth (Davis and Brinson, 1980). The Turbidity Tolerance Index provides an approximation of the relative resilience of macrophyte species to turbidity stress and has been expanded to include more species, including a number of which are known to occur in the UK (Adamus and Brandt, 1990). In addition, species such as charophytes are thought to actively influence the turbidity in their close vicinity (Van den Berg *et al.*, 1997; Van Donk and Van de Bund, 2002; Noges *et al.*, 2003).

Classification schemes

In the Danube Delta Lakes project the lake typology in the Danube delta was described using phytoplankton, hydrology, chemistry, macrophytes and fish species based on PCA analysis (Oosterberg *et al.*, 2000). Here, three groups of lakes could be characterised by their macrophyte communities by correlating lakes with macrophytes and environmental factors, including nutrients. Coops *et al.* (1999) also performed a classification for the Danube delta based on aquatic vegetation and turbidity. For the WFD a number of macrophyte classification schemes have recently been reported for Denmark (Søndergaard *et al.*, 2003), the Netherlands (Van den Berg *et al.*, unpublished), Finland (Kanninen *et al.*, 2003), Germany (Schaumberg *et al.*, 2004), Northern Ireland (Dodkins *et al.*, 2003) and the UK (Willby, pers. comm.). All these macrophyte classification schemes share the common characteristic of not being exclusively related to nutrients. Instead, most macrophyte classification schemes are indirectly related to nutrient conditions. Examples are given in Table 1.3.

As no true direct relationship between transparency and macrophytes has been established, no classification schemes for this parameter have been developed. Most classification schemes have concentrated on the parameters that cause changes in transparency, and so are indirectly related. In the case of nutrient pressures these have typically focused on the association of macrophyte species with variables that correlate with trophic status (e.g. Palmer *et al.*, 1992 - TRS and alkalinity). The general expected trend in European lakes is that transparency will decrease with increasing productivity of the water (Ellenberg, 1988; Palmer *et al.*, 1992). The expected turbidity states for various lake trophic types are as follows (see also Figure 1.4):

- Dystrophic lakes – coloured water
- Oligotrophic (lime deficient) lakes – generally clear water

Table 1.3 Published examples of lake macrophyte models or classification schemes indirectly related to nutrient conditions.

Reference	Region	Notes
Palmer *et al.* (1992)	UK	The Trophic Ranking Scheme (TRS) used the macrophyte composition recorded in 1224 standing waters to classify UK lakes into 10 vegetation groups, which were related to lake alkalinity, pH and conductivity. Each of these 10 groups were allocated site types based on lake trophy. Individual macrophyte species were also allocated a TRS based on the range of site types within which they were found. The average site TRS can be used to infer whether eutrophication has occurred. However, the classification scheme is not reference based and does not directly relate macrophyte species to nutrient conditions.
Swedish EPA (2000)	Sweden	The Swedish Environmental Quality Criteria (SEQC) scheme assesses the state of lakes using a variety of factors including nutrients and species richness of macrophytes. Comparisons of the current condition with reference values are used in appraisals. The SEQC scheme defines conditions for both nutrient loadings and macrophytes at high to low ecological status as well as deviations from the high reference state. The macrophyte scheme is based on the UK TRS (Palmer *et al.*, 1992) and does not, however, directly relate the aquatic plant communities to nutrient concentration data.
US EPA (1998)	USA	The US EPA Lakes and Reservoir Bioassessment and Biocriteria uses submerged macrophytes as one of 7 biological monitoring elements for assessing the condition of US lakes. Lake condition is assessed using additive indices that integrate both habitat and biological scores. The LRBB scheme provides reference values for macrophyte metrics and nutrients but doesn't directly relate them together. This multimetric index indicates the overall biological condition of a lake, however, it cannot quantify the actual cause of degradation, although it does suggest where eutrophication may be the cause.
Heegaard *et al.* (2001)	Northern Ireland	The Northern Ireland Lake Survey quantified macrophyte species – environmental relationships from over 500 lakes using Generalised Additive Models (GAM) and Canonical Correspondence Analysis (CCA). Nutrient concentrations appeared influential in 'explaining' species distribution, but were highly correlated with alkalinity and altitude.
Copps *et al.*, (1999)	Romania	A classification of Danube Delta lakes based on aquatic macrophytes and turbidity.
Moss *et al.* (2003) Sondergaard *et al.* (2003) Kanninen *et al.* (2003) Schaumberg *et al.* (2004) Dodkins *et al.* (2003)	Europe Denmark Finland Germany Northern Ireland	Preliminary macrophyte classification schemes developed specifically for WFD.

- Oligotrophic (lime rich(marl)) lakes – very clear water
- Mesotrophic (macrophyte dominated) lakes – generally clear water although may have seasonal turbidity due to phytoplankton
- Mesotrophic (phytoplankton dominated) lakes – turbid water due to phytoplankton
- Eutrophic lakes – turbid water due to phytoplankton and suspended solids.

In dystrophic lakes the clarity of water is most likely to be as a result of peat colouring rather than as a result of turbidity arising from phytoplankton growth resulting from increased nutrient concentrations. Shifts between these broad lake trophic types would indicate changing status but there is insufficient data available to establish more explicit working ranges for transparency.

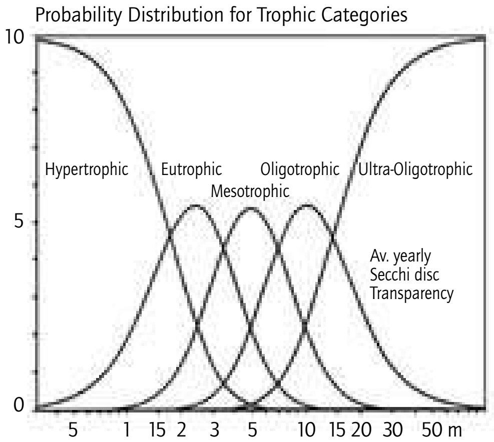

Figure 1.4 Relationship between lake trophic category and transparency (Secchi disc dept; OECD, 1982).

Models

Detailed modelling studies have been carried out examining macrophyte attributes in relation to environmental factors (e.g. Rørslett, 1991; Scheffer *et al.*, 1993; Weisner *et al.*, 1997; Muhammetoglu and Soyupak, 2000; Aseada *et al.*, 2001; Murphy, 2002; Murphy and Hootsmans, 2002). Linear regression models and Generalised Additive Models have been employed to try and model the response of individual macrophyte species to environmental conditions (Heegaard *et al.*, 2001; Van den Berg *et al.*, 2003), or the interaction between species (Van Nes, 2002). These models may have relevance for WFD bioassessment in linking macrophyte community attributes to nutrient conditions.

However, as yet, no reliable quantitative models relating transparency and lake macrophyte composition have been developed. However, it should be noted that in the Netherlands Schaminee *et al.* (1995) described the vegetation types in the aquatic environment and included the main abiotic characteristics. Bal *et al.* (1995) also provided detailed information of species composition in relation to abiotic conditions. These paved the way for the development for the Dutch WFD macrophyte classification scheme.

1.4 Phytobenthos

Introduction

Phytobenthos are an important component of primary production in lakes, although its contribution to overall primary production decreases with increasing lake depth. In deep lakes, the phytobenthos is, however, still an important component of littoral food webs, and will influence the structure of littoral macrophyte, invertebrate and fish communities. For example, an increasing abundance of epiphyton, in response to nutrient enrichment, has long been considered an important factor in the loss of submerged macrophytes (Phillips *et al.*, 1978; Sand-Jensen and Søndergaard, 1981) and the phytobenthos itself constitutes a significant source of energy for most littoral invertebrate grazers.

Despite its potential significance, phytobenthos has received relatively little attention in terms of its use as an indicator of lake quality. The fact, that phytobenthos respond to both water column nutrient concentrations and habitat quality, is accessible from the lake shore, and is less dynamic than the phytoplankton community has led to increasing interest in its use as a monitoring tool for lakes (US EPA, 1998).

Annex V of the Water Framework Directive specifically outlines phytobenthos composition and abundance as two criteria that need defining for type-specific ecological assessment of lakes in relation to undisturbed conditions. Eutrophication of the overlying water generally results in an enhanced growth of attached algae and changes in community composition. Species associated with low phosphorus (the diatoms *Achnanthidium minutissimum* and *Gomphonema tenellum*) and low nitrogen (nitrogen-fixing species, such as the diatoms *Epithemia adnata* and *Rhopalodia gibba* and cyanobacteria such as *Anabaena* spp.) waters have been shown to disappear following nutrient enrichment (Fairchild *et al.*, 1985). If nutrient enrichment proceeds further, phytoplankton can shade the phytobenthos, reducing abundance and shifting community composition to species tolerant of low light levels. The detailed response of the phytobenthos to nutrient conditions is, however, much more complex than for phytoplankton. Nutrients diffuse much more slowly into attached communities, with strong gradients through boundary layers and the attached community. For example, denitrification by bacteria within biofilms can result in nitrogen limitation of the phytobenthos irrespective of nutrient availability in the water column. Further difficulties, in relating water column nutrient concentrations to benthic algae, is that they can also obtain nutrients from the substrates that they are attached to.

Phytobenthos abundance is generally measured as chlorophyll-*a* per unit area of substrate. However, quantitative analysis of phytobenthos abundance is much more difficult than for phytoplankton, due to problems in standardising sampling area and the greater proportion of detrital material within phytobenthos communities.

Only recently have attempts been made to develop quantitative relationships between the phytobenthos community and nutrient conditions. Danilov and Ekelund (2000) analysed epiphyton and epilithon species diversity from seven Swedish lakes. They concluded that epiphyton diversity showed little relationship with nutrient concentrations, but epilithon diversity was consistently related and could be used as an indicator of nutrient status. King *et al.* (2000) examined distributions of 138 epilithic diatom species from 17 lakes in the English Lake District and showed that total phosphorus and calcium concentrations were the most important variables explaining species distributions.

Classification and monitoring schemes

No lake classification schemes based on phytobenthos composition or abundance have been fully developed. Schemes are under development for the WFD in several Member States, although details have only been published for Germany (Schaumburg *et al.*, 2004). Phytobenthos is recommended as one of seven potential biological parameters in Tier 2 of the US EPA lake monitoring and classification scheme (US EPA, 1998). The scheme highlights the potential of phytobenthos, but does point out that responses to pollution or disturbance are not adequately known and require further development.

Surface sediment diatom assemblages are important members of the phytobenthos community. Direct, quantitative relationships between diatom species and total phosphorus concentrations have been developed across Europe (Bennion, 1994; Wunsam and Schmidt, 1995). The combined response of phytoplankton and phytobenthos communities is probably the most established representation of lake ecosystem response to eutrophication and would be highly compatible with palaeolimnological methods for the setting of reference conditions.

Models

No models have been established associating phytobenthos composition or abundance in lakes to specific hydro-morphological or physico-chemical parameters.

1.5 Benthic invertebrates

Introduction

Eutrophication is associated with an increased mineral nutrient supply, mainly N and P, which usually results in an increase in primary production (Sas, 1989). This eventually leads to an increase in the sedimentation of dead plankton (Baines and Pace, 1994) with a resultant increase in decomposition by micro-organisms. Increased decomposition is accompanied by enhanced oxygen consumption by micro-organisms, which can result in oxygen depletion, particularly at the bottom sediments (Dinsmore and Prepas, 1997) and in the profundal zone of deep lakes. This indirect effect of eutrophication on oxygen conditions can lead to significant changes in the structure of oxygen dependant communities, either as a change in dominant species with mild enrichment or severe shifts in species composition with more severe enrichment. The most sensitive communities to changes in oxygen conditions are faunal communities.

The benthic invertebrates are frequently used as indicators of the biological integrity of ecosystems because they are found in most aquatic habitats. They are a diverse and generally abundant group with a wide range of environmental tolerances and preferences. They therefore serve as a useful tool for detecting long-term environmental perturbation resulting from point and non-point sources of pollution (Cairns and Pratt, 1993).

Benthic invertebrates play an essential role in key processes of lakes - in food chains, productivity, nutrient cycling and decomposition. In principle, any environmental changes in lakes, for example in nutrient concentrations, would be reflected by changes in the structure of the benthic invertebrate community. Profundal macroinvertebrates form an important link between detrital deposits and higher trophic levels in aquatic food webs (Brinkhurst, 1974).

Food availability has been postulated as a key factor for benthic communities, and the coupling of pelagic and benthic production has been demonstrated in lake Esrom (Johnasson, 1996) and lake Erken (Goedkoop and Johnson, 1996). However, in other freshwater bodies oxygen content has proved to be the limiting factor for the benthos, especially when the anoxic period lasts more than 4 months (Heinis and Crommentuijn, 1992).

Historically most studies have focused on the changes of the profundal benthic invertebrate communities of deep stratifying lakes (mean depth greater than 3 m). In these deeper waters, below the photic and wave action zones, environmental conditions tend to be relatively uniform and predictable. In contrast, a greater variety of invertebrates are found in the more complex and diverse habitat conditions of smaller shallow (mean depth less than 3 m), non-stratified lakes or littoral areas of deep lakes.

The larvae of the midge genus *Chironomus* (chironomids belonging to the family Chironomidae) are common in the profundal zones of lakes and reservoirs around the world (Armitage *et al.*, 1995). The capacity of these insects to live at very low oxygen concentrations is well known. Chironomids are probably the most useful profundal indicator group for the trophic state of a lake, as they have high species richness compared to other benthic invertebrate groups. They occur over the whole spectrum of nutrient conditions and, because individual species have highly specific environmental tolerances, they change species composition in tandem to changing lake trophic status (Rosenberg, 1992). Multiple regression analyses of data from Spanish lakes and reservoirs, showed that Chironomus density decreased with depth and temperature and increased with alkalinity (Real, 2000).

Species indicative of eutrophic conditions will gradually colonize a lake (in most cases species belonging to class Oligochaeta and family Chironomidae). Although, freshwater worms, oligochaets (Oligochaeta) are more sensitive to changes in oxygen conditions than chironomids, because they are less mobile and they feed and reproduce deeper in the sediments than the latter (Lang, 1999).

Annex V of the WFD outlines the following benthic invertebrate quality components that need consideration in the assessment of the ecological status of lakes:

- Composition and abundance of benthic invertebrate fauna
- Ratio of disturbance sensitive taxa to insensitive invertebrate taxa
- Diversity of invertebrate communities.

In the WFD, a declining ecological quality is associated with poor diversity and dominance of a few taxa. To comply with 'good ecological status' the composition and abundance of invertebrate taxa, the ratio of disturbance sensitive taxa to insensitive taxa and levels of diversity should show only *slight* signs of alteration compared to the type-specific reference communities (Annex V, Section 1.2.2).

Metrics

Taxonomic composition. The tolerances of profundal macroinvertebrates to low dissolved oxygen (DO) concentrations vary widely among species (Davis, 1975; Herreid, 1980). Attempts to devise a lake typology for macroinvertebrates in general and specifically for benthic chironomid assemblages, based on trophic status and DO concentration, have met with varying degrees of success (Brinkhurst, 1974), although clear general patterns exist. A combination of both sediment type and oxygen supply have been shown to control the distribution of chironomids (Francis and Kane, 1995) and oligochaetes (Verdonschot, 1996). The Orthocladiinae are characteristic of highly oxygenated waters and thus tend to reflect high status as does a high ratio of the oligochaete Naididae to chironomids (Armitage *et al.*, 1995). Common compositional metrics include the percentage of individuals of the family Orthocladiinae among chironomid larvae present and the ratio between the number of individuals of the oligochaete Naididae to the sum of all specimens of Naididae and chironomids (Moss *et al.*, 2003). In contrast, particularly in shallow lakes, other authors have demonstrated that biotic factors, rather than environmental factors, can be responsible for the composition of the benthic macroinvertebrate community. The biotic factors examined were mainly the effect of predation (Kornijow, 1997) but also intra- and interspecific competition (Van de Bund and Groenendijk, 1994).

Barbour *et al.* (1995) recommend relative (percentage) rather than absolute abundance in calculating metrics as they have been shown to be more robust and reliable and were more likely to reflect structural changes resulting from nutrients. An additional composition attribute is Bray-Curtis dissimilarity, a coefficient shown to be a robust and ecologically interpretable index of changes in taxonomic composition (Legendre and Legendre, 1998).

Benthic invertebrates abundance. Increased phytoplankton production associated with eutrophication and the concurrent increase in organic sedimentation results in decreasing oxygen concentration at the sediment/water interface. For benthic invertebrates, this means more food but reduced oxygen availability. As a result, characteristic species indicative of eutrophic conditions gradually colonise (in most cases oligochaetes and chironomids). Oligochaetes are more affected by enrichment than chironomids as they are less mobile and they depend more on the inner sediment for their food and reproduction than the latter (Lang, 1999).

In lakes with a strong oxygen gradient, profundal macroinvertebrate biomass (PMB) typically declines from the upper profundal region to the maximum depth (Brinkhurst, 1974). Conversely, the artificial enhancement of hypolimnetic DO concentrations in productive lakes, through aeration and oxygenation treatments, is often followed by a dramatic increase in profundal macroinvertebrate abundance (Dinsmore and Prepas, 1997). Rasmussen and Kalff (1987) hypothesised that the observed negative correlation of PMB with high phytoplankton biomass in their regression models truly reflected the deleterious effects of decreased hypolimnetic DO concentration on profundal macroinvertebrates. In addition, Takamura and Iwakuma (1990) suggested that extremely high primary productivity leading to anoxic conditions at the sediment/water interface may depress PMB even in shallow polymictic lakes.

Benthic invertebrates diversity. Measures of diversity have only occasionally been used in studies of the profundal benthos primarily due to the difficulty in identifying taxa and because profundal communities are difficult to characterise using diversity indices as they tend to be dominated by comparatively few species (Wiederholm, 1980). Wiederholm (1980) suggested that species richness or the total number of taxa is a much better measure of the diversity of the benthos by demonstrating that a strong correlation could be made between species richness (once adjusted to sample depth) and the trophic state (as indicated by chlorophyll-*a* concentrations). Richness and diversity attributes includes the total number of taxa (richness per unit area) and numerical richness (Larsen and Herlihy, 1998). Low diversity indices are associated with eutrophicated ecosystems.

Classification schemes

Benthic invertebrates are widely utilised as indicators of organic pollution in freshwaters, particularly rivers due to their sensitivity to oxygen conditions (Hynes, 1960; Hellawell, 1986). Early classification schemes in lakes, similar to the Saprobic systems in rivers, were based mainly on the analysis of the species composition of the chironomid communities. See Table 1.4 for examples.

The Benthic Quality Index (BQI) was developed for assessing trophic status of Palearctic lakes (Wiederholm, 1980). In these ecosystems, a eutrophic lake's BQI value is considered to be 1 and *Chironomus plumosus* the dominant taxon; a BQI value of 5 is characteristic of an oligotrophic lake. If no indicator species are present then a value of 0 is recorded and is indicative of a hypereutrophic lake (Wiederholm, 1980). Lang developed three indices of trophy for lake Geneva based on the structure of the Tubificidae and Lumbriculidae communities (Lang, 1998).

Numerous other studies highlight oxygen concentrations and food availability (as indicated by total phosphorus, chlorophyll-*a* and algal biovolume) to be the most important community-structuring factors linking chironomid communities with lake trophic state (e.g. Saether, 1979; Brodin, 1982). In northern temperate waters these models have proved to be useful (e.g. Kansanen *et al.*, 1984) as long as oxygen concentrations in the profundal zone remained high enough to support the chironomid communities (Brodersen and Lindegaard, 1999).

Table 1.4 Examples of benthic invertebrate classification systems and models.

Reference	Region	Notes
Wiederholm (1980)	Palearctic	Benthic Quality Index (BQI) used to assess trophic status of Palearctic lakes. In this system in eutrophic lakes the BQI value is considered to be 1 and *Chironomus plumosus* the dominant taxon; a BQI value of 5 is characteristic of oligotrophic lakes. If no indicator species are present then a value of 0 is scored indicating a hypereutrophic lake.
Lang (1998)	Lake Geneva, Switzerland	Indices of trophy were developed based on the structure of the Tubificidae and Lumbriculidae (Oligochaeta) communities.
Brodin (1982) Heinis and Davids (1993) Saether (1979) Kansanen *et al.* (1984) Lindegaard (1995) Brodersen and Lindegaard (1999)	Sweden The Netherlands Norway Finland Denmark	Studies linking chironomid communities with lake trophic state (indicated by variables such as oxygen concentration, total phosphorus, chlorophyll-*a* and algal biovolume).
Dinsmore *et al.* (1999)	Alberta, Canada	Empirical model linking profundal macroinvertebrate biomass (PMB) and water chemistry and morphometric variables from 26 lakes located within the Boreal Mixedwood and Boreal Subarctic ecoregions.
King and Richardson (2003)	Florida, USA	Five macroinvertebrate core metrics were linked to surface-water TP from an observed P gradient and a P-dosing experiment in coastal wetlands of the south Florida to estimate numerical water quality criteria for TP.
Astrakhantsev *et al.* (2003)	Lake Ladoga, NW Russia	Model of the annual dynamics and distribution of zoobenthos biomass with respect of lake phosphorus dynamics

Models

Empirical models of profundal macroinvertebrate biomass using dissolved oxygen concentration and water temperature as predictors have explained a significant proportion of the variance found in profundal macroinvertebrate biomass (PMB; Dinsmore *et al.*, 1999). Residual variation in this model is high suggesting that the model could be improved by obtaining more representative estimates of PMB and water chemistry variables; including estimates of detrital production, accumulation rate and composition; or by quantifying the effects of biotic factors such as competition, predation, life history phenologies[1], which have an important influence on PMB in habitats where dissolved oxygen and food materials are not limiting (Kajak, 1987).

Empirical models have established relationships between total phosphorus and the structure and function of macroinvertebrate communities (e.g. King and Richardson, 2003). A model developed for Lake Ladoga, North-Western Russia, reproduced the annual dynamics of zoobenthos biomass and its distribution over the lake bed (Astrakhantsev *et al.*, 2003).

1.6 Fish

Introduction

According to the cascading trophic interaction hypothesis a top predator such as piscivorous fish can influence the abundance, size-structure, and productivity of zooplankton and phytoplankton (Carpenter and Kitchell, 1984; Carpenter *et al.*, 1985). The bottom-up and top-down model (McQueen *et al.*, 1986; McQueen *et al.*, 1989) combines the influence of both predators (top-down) and resources (bottom-up). It predicts that top-down forces should be strong at the top of the food web and weaken towards the bottom, whereas bottom-up forces should be strong at the lower trophic level (algae) and weaken towards the top of the food chain (little or no influence of fish). Top- down pressure can exert substantial influence on phytoplankton composition and biomass through grazing pressure, however the importance depends on the number of food web links (2 to 5) and trophic status of the lakes (oligo- to hyper-trophic).

Due to their complex ecological requirements, fish are sensitive indicators for habitat quality at various spatial scales. As consumers and/or top predators, they integrate information on trophic conditions across the food chain. They also provide detailed information on the respective trophic level. There is a long tradition of linking the health of fish populations to water quality.

Dissolved oxygen is a defining and limiting parameter for fish, affecting survival, growth, spawning, swimming performance, larval development and migration behaviour (Doudoroff and Shumway, 1970). Since oxygen is vital to other aquatic biota (i.e. invertebrates) that are a key food source for fish, then the species composition and biomass of fish communities can also be affected indirectly by oxygen availability. The composition of the fish community at a specific site is the result of various factors including environmental factors such as temperature, oxygen, flow and nutrients and the anthropogenic movement of fish.

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[1] Periodic life history events, such as breeding, migration, etc., which occur in relation to seasonality and climate.

Fish species can be separated by their oxygen requirements. As well as having different requirements between species, they have different needs throughout their life stages (eggs, larvae, juveniles and adults). Generally, stenotherms[2] need higher oxygen concentrations than eurytherms[3] and adult fish tolerate low oxygen concentrations better than juveniles. The temperature of the surrounding water has a substantial influence on the dissolved oxygen concentration within the water column, as well as significantly affecting the physiological demand by fish. Warmer water temperatures reduce the oxygen saturation capacity, as well as increasing the physiological demand by the fish. The development of anoxia in lakes is most pronounced in thermally stratified systems in summer and under the ice in winter when the water mass is cut-off from the atmosphere. Besides the direct effects on aerobic organisms, anoxia can lead to increased release of phosphorus from sediments that can fuel algal blooms when mixed into the upper euphotic (illuminated) zone. It also leads to the build-up of chemically reduced compounds such as ammonium and hydrogen sulphide (H_2S) which can be toxic to bottom dwelling organisms. In extreme cases, sudden mixing of H_2S into the upper water column can cause fish kills.
The Water Framework Directive requires information on fish communities to be used in the ecological quality assessment of lakes. Annex V of the WFD outlines three fish-related quality elements that need consideration:

- fish composition
- fish abundance
- fish age structure.

The WFD normative definitions of ecological status classification, indicate that: 1) changes in species composition and abundance; 2) decreases in type sensitive species; and 3) changes in age structure, as a sign of disturbance indicating a failure of reproduction or development of a particular species; will downgrade water quality classification.

Classification schemes

Each fish species has characteristic requirements for water quality, habitat and environmental conditions necessary to satisfy their need for breeding, feeding, growing, recruitment and survival. These characteristics are used to classify fish species according to the concept of ecological/functional guild, defined as a group of species that use the same class of environment conditions in the same way (Root, 1967; Austen *et al.*, 1994). Examples of classification schemes are summarized in Table 1.5.

The European Fish-Index (EFI) developed by the FAME project[4], is a multimetric fish-index for classification of European rivers. The EFI is based on the same principle of the Index of Biotic Integrity (IBI; Karr, 1981), where the fundamental assumption is that the composition and structure of fish assemblages are impacted by human pressures in a predictable manner. In the EFI the fish species were classified according to their zoogeographic status (Native, Introduced,

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[2] Organisms adapted** to only slight variations in temperature.
[3] Organisms adapted*** to a wide range of temperature.
[4] EVK1-CT-2001-00094; http://fame.boku.ac.at.

Table 1.5 Examples of Fish-based classification systems for lakes.

Reference	Region	Notes
Jeppesen *et al.* (2000)	Denmark	Fish species richness, biodiversity and trophic structure are linked to a trophic gradient of TP
Moss *et al.* (2003)	Europe	Classification system developed for 48 European lake ecotypes using 28 variables for water quality status with 3 fish parameters: fish community, fish biomass and piscivorus-zooplanctivorus biomass ratio
Regier *et al.* (1998) Navodaru *et al.* (2003) Oosterberg *et al.* (2000)	Europe Danube Delta lakes	Classification of fish communities in relation with a lake ecology guild, such as riverine/ white fish, euritopic/ grey fish and limnophilic/ black fish.
Ligtvoet and Grimm (1992)	The Netherlands	Occurrence of three fish communities (perch-type, pike-perch-type, and pike –perch-bream-type) and the size of total fish stock, perch stock, pike stock and cyprinid stock linked with summer average TP concentration.
Gassner *et al.* (2003)	Austria	A relationship between TP ($mg.m^{-3}$) and total fish biomass in $kg.ha^{-1}$ were calculated for prealpine oligo- and oligo-mesotrophic lakes using: $BM = 3.8148 \times TP^{1.0940}$, $r^2 = 0.59$, n = 10.
Carvalho *et al.* (2002) EIFAC (1973) US EPA (1986; 1997)	UK Europe USA	Classification criteria between threshold requirement for oxygen concentrations and fish communities (e.g. in UK: dissolved oxygen >8$mg.l^{-1}$ for salmonids, >6 for non-salmonids and >1 for tolerant species of cyprinids). Similar or slightly different thresholds have been forwarded by the European Inland Fishery Advisory Commission (EIFAC), National Research Council of Canada (NRCC), and US Environmental Protection Agency (US EPA).
Swedish Environmental Protection Agency (www.internat.naturvardsverket.se)	Sweden	Method for classifying fish communities in rivers and lakes using parameters from fish surveys and physical characteristics of the water bodies. The classifications are not directly related to specific physico-chemical parameters, but aim to define the extent to which these communities may deviate from 'undisturbed' waters.
Busnita and Brezeanu (1967) Nikolski (1962)	Romania	Classification systems based on fish ecology and oxygen sensitivity.
Diudea *et al.* (1986)	Romania	Direct nutrient stress on fish. Toxic effects of nitrite and nitrate, depend on species, salinity, concentration and time of exposure. Maximum level of Romanian STAS range of 10-45 $mg.l^{-1}$ for nitrate ($-NO_3$) and 0-1 $mg.l^{-1}$ for nitrite ($-NO_2$), depending of water category use.

Endemic), trophic guild, reproductive guild, habitat guild (degree of rheophily[5] and habitat preference in the water body), migratory behaviour, longevity and tolerance capacity (Schmutz *et al.*, in prep.). Where possible, tolerance to specific stressors was assessed (temperature, acidification), including habitat degradation and water quality. Where information was lacking an overall assessment of general tolerance was made. A group of sentinel species, i.e. known to be common in specific river-zones and providing good information on ecological quality, was identified for

[5] Degree of preference for running waters.

each ecoregion. For these species, information regarding size or age structure is required in the multimetric assessment. The EFI indicates the deviation from predicted reference metrics, and provides a probability for a site to represent reference conditions.

Final ecological status of a site is identified by the degree of deviation from reference metrics. The EFI is able to detect impacts of both physical and chemical pressures. However more precise data on pressures are required to distinguish between different types of pressure. The EFI index cannot be applied directly for fish biota in lakes, but the same approach could be applied to develop a fish-index for lakes.

A Fish-based classification system has been developed for Danish lakes (Jeppesen *et al.*, 2000). Changes in fish species richness, biodiversity and trophic structure were linked to a trophic gradient which was divided into five total phosphorus (TP) classes (class 1-5: <0.05, 0.05-0.1, 0.1-0.2. 0.2-0.5, >0.4 mg P L^{-1}). The fish species richness was unimodally related to TP, being highest at 0.1-0.4 mg P L^{-1}. At low nutrient concentrations, piscivorous fish (particularly perch, *Perca fluviatilis*) were abundant and the biomass ratio of piscivores to plankti-benthivorous cyprinids was high and density of cyprinids low. With increasing TP, a major shift in trophic structure occurred.

A classification system developed for 48 European lake ecotypes by the research project ECOFRAME proposed 28 variables for water quality status from which three were fish parameters: fish community, fish biomass and piscivorus-zooplanctivorus biomass ratio (Moss *et al.*, 2003). The advantages of the proposed classification system are that it applies variables that are inexpensive to measure and ecologically relevant. The scheme was tested using data from 66 shallow lakes, and may be adapted for more lakes after minor adjustments.

A classification of fish communities into lake ecology guilds, such as riverine/ white fish, eurytopic/ grey fish and limnophilic/ black fish has been proposed for Europe. "White fish" are Acipenseridae, Salmonidae and Clupeidae, "grey fish" Percidae and Esocidae and "black fish" Umbridae and Cobitidae; representatives of Cyprinidae turn up in all three classes (Regier *et al.*, 1998). Except for diadromous[6] migratory species, all these categories are found in lakes of the Danube delta that are connected to a river with a network of natural or man-made canals (Navodaru *et al.*, 2002). Abundance, biomass and species guild was used to describe a trophic gradient of Danube delta lakes. PCA analysis was carried out to separate lake types for the Danube delta using comprehensive data sets of hydrology, chemistry, vegetation, phytoplankton, and fish (Oosterberg *et al.*, 2000). Based on this study three types of lake were identified: Type I, mostly eutrophic with intermediate fish species (grey and black), Type II, an intermediate with eurytopic/grey fish and Type III, mesotrophic with limnophilic/black fish (Figure 1.5).

Models

Some examples of models for fish and environmental quality variables for lakes are reported in Table 1.6. A study of 466 lakes from temperate to artic zones described the impact of nutrients and lake depth on top-down control in the pelagic zone of lakes (Jeppesen *et al.*, 2003). The study

[6] Fish species migrating between fresh and marine waters.

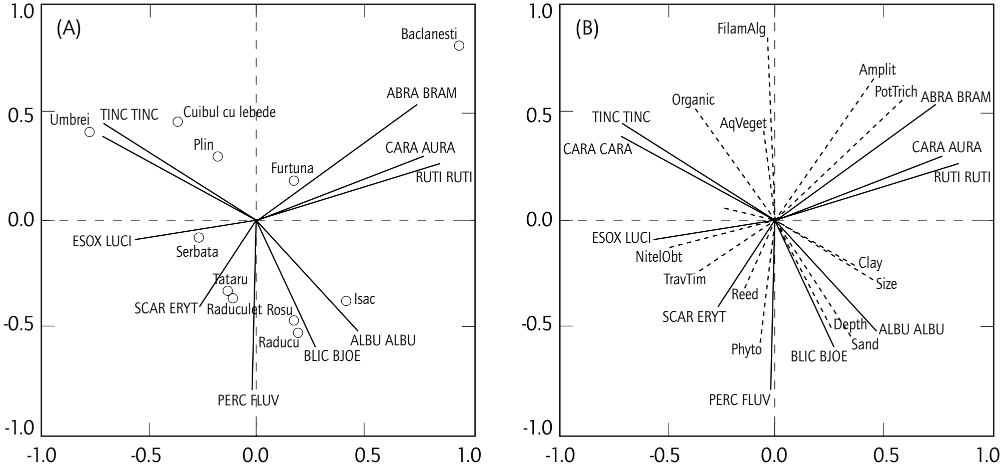

Figure 1.5 PCA analysis of the relative biomass of 10 dominant fish species in 11 lakes in relation to trophic variables, (A) species-site bi-plot of the first two axes, and (B) species-environment bi-plot of the first two axes. Abbreviations of fish species are composed by the first 4 letters of both genus and species names. Soil type (ORGANIC, SAND or CLAY), morphometry (SIZE, DEPTH), hydrological distance (TRAVTIM) together with subsequent residence time in the lake (CURESTM), water level fluctuation (AMPLIT), chlorophyll a concentration (PHYTO), filamentous algae (FILAMALG), aquatic vegetation coverage (AQVEGET), Potamogeton trichoides (POTTRICH), Nitellopsis obtusa (NITELOBT) and reed: open water ratio (REED). (original from Navodaru *et al.*, 2002).

concluded that (a) fish predation pressure on large-bodied cladocerans is overall unimodally related to TP in both shallow and deep lakes; (b) the effect generally cascades to the phytoplankton biomass level in eutrophic lakes, but less significantly in oligotrophic lakes; (c) predator control of large-bodied zooplankton tends to be higher in shallow lakes than in deep lakes. A multiple regression model revealed that the percentage abundance of *Daphnia* was significantly inversely related to Catch Per Unit Effort (CPUE) and positively related to lake water TP, suggesting a higher predation risk at a given CPUE in oligotrophic lakes.

Fish yield potential (FYP) was estimated for 786 lakes in north-east Germany using data on primary production of phytoplankton (PP) and total phosphorus (TP) as the main variables (Brämick and Lemcke, 2003). The following equation between TP and PP was empirically derived from 8 stratified German lakes:

$$PP\ (g\ C.m^{-1}a^{-1}) = 148\ \log TP(\mu g.l^{-1}) - 39.6$$

A relationship between TP in stratified and shallow lakes, that had similar trophic status, gave the following equation:

$$TP_1 = 1.48198\ TP_2.^{1.2278}$$

where TP_1 = TP in stratified lakes, and TP_2 = TP in shallow lakes. The trophic status for both types of lakes was obtained from the German States Association for Water (1998), where the trophic state of a lake is based on four parameters, including TP-value during the spring turnover. Based

Table 1.6 Examples of models for fish and environmental quality variables for lakes.

Reference	Region	Notes
Jeppesen *et al.* (2003)	Denmark, Norway (temperate, boreal and artic zones)	Multiple regression model – data from 466 lakes was used to study impacts of lake trophic status on top-down control by fish. The results indicated that planktivorous fish have a more limited effect on the grazing control of phytoplankton in oligotrophic lakes than in eutrophic lakes, despite similar predator control of large-bodied zooplankton.
Hanson and Legged (1982)	Canada	Linear model for relationship between total phosphorus (TP) and fish yield (Y) in two lakes: LogY = 0.708 log TP + 0.774 Log Y = 0.072 TP + 0.792
Brämick and Lemcke (2003)	NW Germany	Data from 786 lakes was used to estimate fish yield potential (FYP; $kg.ha^{-1}.a^{-1}$) for shallow and stratified lakes as function of primary production of phytoplankton (PP) and total phosphorus (TP) lakes with PP<380: FYP= $6.315e^{0.0062.pp}$, r=0.47; lakes with PP>380: FYP= 57.937 lnPP – 278.09, r = 0.34
Lyytikainen and Jobling (1998)	Northern Finland	Study from an Artic lake Inari, indicated that an increase in lake temperature (T) resulted in an immediate increase in oxygen consumption (M-accl in mg kg^{-1} h^{-1}) fitting to an exponential model: M-accl=$46.53e^{(0.086T)}$

on these models, Brämik and Lemcke (2003) developed regional fish yield estimation models for shallow and stratified lakes. The models take into account the findings by Lang *et al.* (1981), who state that the relation between PP and FYP changes from an exponential to a logarithmic function with increasing PP values and approaches the upper asymptote. Accordingly, the following models were obtained (Brämick and Lemcke, 2003):

$$\text{FYP (for lakes with PP < 380): FYP}(kg.ha^{-1}.a^{-1}) = 6.315e^{0.0062.pp},\ r = 0.47;$$

$$\text{FPY (for lakes with PP > 380: FYP}(kg.ha^{-1}.a^{-1}) = 57.937 \ln PP - 278.09,\ r = 0.34$$

These models are capable of simulating the effects of a large variety of environmental conditions, and can be used as dynamic tools for ecosystem risk assessment since they produce both qualitative and quantitative results, allowing for comparisons of predictions with on-going observational research and ecosystem monitoring.

1.7 Summary

Although the impact of nutrient pressures on biological quality is relatively well understood for lakes in qualitative terms, there has been very limited development of quantitative dose-response relationships, classification tools or models. This review identified a number of widely-recognised strong relationships between physico-chemical measures of eutrophication and associated biological responses. These can be split into two types of relationships: (1) primary responses (phytoplankton, phytobenthos and macrophytes) to nutrient state, and (2), secondary responses to primary production or production-related decreases in transparency and oxygen. The advantages and disadvantages associated with developing these elements as indicators of eutrophication pressures are briefly summarised below.

Primary responses to nutrients. Phytoplankton abundance (most often measure of abundance taken: chlorophyll-*a*) is highly sensitive to nutrients and integrates the response to a range of possible limiting nutrients (P, N, Si). It is also a simple, practical measure of eutrophication impacts that is widely adopted across EC Member States. Phytoplankton composition is sensitive to nutrients and qualitatively well understood. The response to nutrients is, however, complicated by alkalinity, stratification and lake morphometry. Macrophyte composition is sensitive to nutrients and qualitatively well understood. Historical records are frequently available for defining reference conditions. The response to nutrients is, however, affected by substrate quality and water level fluctuations (hydro-morphological pressures).

Secondary responses to nutrients. Fish biomass is sensitive to nutrient-related primary production, but limited data are available. The relationship between phytoplankton abundance and transparency is a simple, practical measure of eutrophication impacts that is widely adopted across EC Member States, but it is affected by water colour and clay particles. The relationship between macrophyte composition/abundance and transparency is qualitatively well understood, but limited data are available. The relationship between benthic invertebrate composition and oxygen is sensitive to eutrophication pressures but limited data are available. The relationship between fish composition and oxygen is qualitatively well understood and sensitive to eutrophication pressures but limited data are available.

All these primary and secondary responses are likely to have robust relationships with eutrophication pressure, but data are limited for some of these parameters compared with others, which puts practical limits on their selection as indicators for WFD ecological quality assessment. In particular, data on phytobenthos, benthic invertebrates and oxygen conditions in lakes are very limited, as are data on macrophyte and fish abundance. Although, some relationships, such as fish composition – oxygen condition, are relatively well understood, despite the data limitations. This calls for an intensified effort to develop indicators in order to allow implementation of the WFD. Availability of indicators and classification metrics is not only restricted by data, but also by the limited research effort to develop Europe-wide bio-indicators for all the quality elements required to be assessed by the WFD.

1.8 References

Adamus, P., and K. Brandt. 1990. Impacts on quality of inland wetlands of the United States: a survey of indicators, techniques, and applications of community level biomonitoring data. EPA/600/3 - 90/073, US EPA, pp. 406

Anonymous. 2003. Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance No 10, River and lakes – Typology, reference conditions and classification systems, pp. 94

Armitage, P.D., P.S. Cranston, and L.C.V. Pinder. 1995. The Chironomidae:Biology and Ecology of Non-Biting Midges. Chapman and Hall, London

Asaeda, T., V.K. Trung, J. Manatunge, and T.V. Bon. 2001. Modelling macrophyte-nutrient-phytoplankton interactions in shallow eutrophic lakes and the evaluation of environmental impacts. Ecological Engineering 16:341-357

Astrakhantsev, G.P., T.R. Minina, N.A. Petrova, V.N. Poloskov, and L.A. Rukhovets. 2003. Modeling Lake Ladoga zoobenthos and studying its role in phosphorus exchange at the water–bed interface. Water Resources 30:209–220.

Austen, D.J., P.B. Bayley, and B.W. Menzel. 1994. Importance of the Guild Concept to Fisheries Research and Management. Fisheries 19:12-20

Baines, S.T., and M. Pace. 1994. Relationships between suspended particulate matter and sinking flux along a trophic gradient and implication for the fate of planktonic primary production. Canadian Journal of Fisheries and Aquatic Sciences 51:25-36

Bal, D.H.M., H.M. Beije, Y.R. Hoogeveen, S.R.J. Jansen, and P.J. van der Reest. 1995. Handboek natuuraeltypen in Nederland. IKC-rapport 11 IKC Natuubeheer, Wageningen

Barbour, M.T., J.B. Stribling, and J.R. Karr. 1995. Multimetric approach for establishing biocriteria and measuring biological condition. In: W.S. Davis, and T.P. Simon (Eds.) Biological assessement and criteria: Tools for water resource planning and decision-making, CRC Press, Boca Raton, pp. 63-77

Bennion, H. 1994. A Diatom-Phosphorus Transfer-Function for Shallow, Eutrophic Ponds in Southeast England. Hydrobiologia, 276:391-410

Blindlow, I. 1992. Decline of charophytes during eutrophication – comparison with angiosperms. Freshwater Biology 28:9-14

Brämick, U., and R. Lemcke. 2003. Regional application of fish yeld estimation procedure to lakes in north-east Germany. Limnologica 33:205-213

Bratli, J.L. 2000. Classification of the Environmental Quality of Freshwater in Norway. In: P. Heinonen, G. Ziglio, and A. Van der Beken (Eds.), Hydrological and Limnological Aspects of Lake Monitoring, John Wiley & Sons, Chichester

Brinkhurst, R.O. 1974. The benthos of lakes. St Martins Press, New York

Brodersen, K.P., and C. Lindegaard. 1999. Classification, assessment and trophic reconstruction of Danish lakes using chironomids. Freshwater Biology 42:143-157

Brodin, Y.Y. 1982. Paleoecological studies of the recent developments of the lake Vaxjosjon: interpretetation of eutrophication process through analysis of subfossil chironomids. Archiv Fur Hydrobiologie 93:313-326

Busnita, T., and G. Brezeanu. 1967. Ihtiofauna. In: Limnologia sectorului romanesc al Dunarii. A.R.S. Romania, Bucuresti, pp. 325-370

Cairns, J. Jr., and J.R. Pratt. 1993. A history of biological monitoring using benthic macroinvertebrates. In: D.M. Rosenberg, and V.H. Resh (Eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates, Chapman and Hall, pp. 10-27

Canfield, D.E. Jr., and L.M. Hodgson. 1983. Prediction of Secchi disc depths in Florida lakes: Impact of algal biomass and organic color. Hydrobiologia 99:51-60

Cardoso, A.C., J. Duchemin, P. Magoarou, and G. Premazzi. 2001. Criteria for the Identification of Freshwaters Subject to Eutrophication. Office for Official Publications: European Commission, Luxembourg, pp. 90

Carlson, R.E. 1977. A trophic state index for lakes. Limnology and Oceanography 22:361-369

Carlson, R.E. 1980. More Complications in the Chlorophyll-Secchi Disk Relationship. Limnology and Oceanography 25:379-382

Carpenter, S.R., and J.F. Kitchell. 1984. Plankton community structure and limnetic primary production. American Naturalist 124:159-172

Carpenter, S.R., J.F. Kitchelland, and J.R. Hodgson. 1985. Cascading trophic interaction and lake productivity. BioScience 35:634-639

Carvalho, L., H. Benninon, A. Darwell, I. Gunn, A. Lyle, D. Monteithand M. Wade. 2002. Physico-chemical conditions for supporting different levels of biological quality for the Water Framework Directive for freshwater. R&D Technical Report to the Environment Agency, December 2002, pp. 145

Carvalho, L., S. Maberly, L. May, C. Reynolds, M. Hughes, R. Brazier, L. Heathwaite, S. Liu, J. Hilton, D. Hornby, H. Bennion, A. Elliott, N. Willby, R. Dils, L. Pope, G. Phillips, and I. Fozzard. 2004. Risk Assessment Methodology for Determining Nutrient Impacts in Surface Freshwater Bodies. R&D Technical Report P2-260/9 to the Environment Agency and Scottish Environment protection Agency, July 2004. pp. 171

Coops, H., J. Hanganu, M. Tudor, and W. Oosterberg. 1999. Classification of Danube Delta lakes based on aquatic vegetation and turbidity. Hydrobiologia 415:187-191

Danilov, R.A., and N.G.A. Ekelund. 2000. The use of epiphyton and epilithon data as a base for calculating ecological indices in monitoring of eutrophication in lakes in central Sweden. Science of the Total Environment 248:63-70

Davis, J.C. 1975. Minimal dissolved oxygen requirements of aquatic life with emphasis on Canadian species: a review. Journal of the Fisheries Research Board of Canada 32: 2295-2332

Davis, G.J., and M.M. Brinson. 1980. Responses of submersed vascular plant communities to environmental change. U.S. Department of the Interior Report FWS/OBS-79/33

Dillon, P.J., and F.H. Rigler. 1974. A test of a simple nutrient budget model predicting the phosphorus concentration in lake water. Journal of the Fisheries Research Board of Canada 31:1771-1778

Dinsmore, W.P., and E.E. Prepas. 1997. Impact of hypolimnetic oxygenation on profundal macroinvertebrates in a eutrophic lake in central Alberta. I. Changes in macroinvertebrate abundance and diversity. Canadian Journal of Fisheries and Aquatic Sciences 54:2157–2169

Dinsmore, P.W., G.J. Scrimgeour, and E.E. Prepas. 1999. Empirical relationships between profundal macroinvertebrate biomass and environmental variables in boreal lakes of Alberta, Canada. Freshwater Biology 41:91-100

Diudea, M., S. Tudor, and A. Igna. 1986. Toxicologie acvatica. Cluj Napoca. pp. 183

Dodkins, I., B. Rippey, and P. Hale. 2003. The advantage of metrics for aquatic macrophyte assessment in Northern Ireland. Temanord report 547:29-34.

Doudoroff, P., and D.L. Shumway. 1970. Dissolved oxygen requirements of freshwater fishes. FAO Technical Paper 86, pp. 291

EIFAC. 1973. Water Quality Criteria for European Freshwater Fish. Report on Dissolved Oxygen and Inland Fisheries. EIFAC Tech. Paper 86, pp. 19

Ellenberg, H. 1988. Vegetation ecology of central Europe. Cambridge University Press, Cambridge

Elliott, J.A., C.S. Reynolds, and A.E. Irish. 2001. An investigation of dominance in phytoplankton using the PROTECH model. Freshwater Biology 46:99-108

Fairchild, G.W., R.L. Lowe, and W.B. Richardson. 1985. Algal Periphyton Growth on Nutrient-Diffusing Substrates - an *In situ* Bioassay. Ecology 66:465-472

Francis, D.R., and T.C. Kane. 1995. Effect of substrate on colonization of experimental ponds by Chironomidae (Diptera). Journal of Freshwater Ecology 10:57-63

Gassner, H., G. Tischler, and J. Wanzenböck. 2003. Ecological integrity assessment of lakes using fish communities - suggestions of new metrix development in two Austrian prealpine lakes. International Review of Hydrobiology 88:635-652

German States Association for Water. 1998. Gewässerbewertung – stehende Gewässer, Berlin, Kulturbuch – Verlag, pp. 74

Gibson, G., R. Carlson, J. Simpson, E. Smeltzer, J. Gerritson, S. Chapra, S. Heiskary, J. Jones, and R. Kennedy. 2000. Nutrient Criteria Technical Guidance Manual: Lakes and Reservoirs. US EPA, Washington DC, pp. 231

Goedkoop, W., and R.K. Johnson. 1996. Pelagic-benthic coupling:profundal benthic community response to spring diatom deposition in mesotrophic Lake Erken. Limnology and Oceanography 41:636-647

Hanson, J.M., and W.C. Legget. 1982. Empirical prediction of fish biomass and yield. Canadian Journal of Fisheries and Aquatic Sciences 39:257–263

Heegaard, E., H.H. Birks, C.E. Gibson, S.J. Smithand S. Wolfe-Murphy. 2001. Species-environmental relationships of aquatic macrophytes in Northern Ireland. Aquatic Botany 70:175-223

Heinis, F., and T. Crommentuijn. 1992. Behavioural response to changing oxygen concentration of deposit feeding chironomid larvae (Diptera) of littoral and profundal habitats. Archiv Fur Hydrobiologie 124:173-185

Heinonen, P. 2000, Water quality classification in Finland. In: P. Heininen, G. Ziglio, and A. Van der Beken (Eds.), Hydrological and limnological aspects of lake monitoring. John Wiley & Sons, Chichester, pp. 345-350

Hellawell, J.M. 1986. Biological indicators of freshwater pollution and environmental management. Elsevier, London

Herreid, C.F. 1980. Hypoxia in invertebrates. Comparative Biochemistry and Physiology 67:311-320

Hillbricht-Ilkowska, A. 1993. Phosphorus loading to lakes of Suwalki Landscape Park (north-eastern Poland) and its relation to lake trophy indices. Polish Journal of Ecology 41:221-235

Hough, R.A., and M.D. Forwall. 1988. Interactions of inorganic carbon and light availability as controlling factors in aquatic macrophyte distribution and productivity. Limnology and Oceanography 33:1202-1208

Hynes, H.B.N. 1960. The biology of polluted waters. Liverpool University Press, Liverpool

Janse, J.H., and L. Van Liere. 1995. PC-Lake, a modelling tool for the evaluation of lake restoration scenarios. Water Science Technology 31:371-374

Jeppesen, E., J.P. Jensen, M. Sondergaard, T. Lauridsenand F. Landkildehus. 2000. Trophic structure, species richness and biodiversity in Danish lakes: changes along a phosphorus gradient. Freshwater biology 45:201-218

Jeppesen, E., J.P. Jensen, C. Jensen, B. Faafeng, D.O. Hessen, M. Sondergaard, T. Lauridsen, P. Brettumand K. Christoffersen. 2003. The impact of nutrient state and lake depth on top-down control in the pelagic zone of lakes: a study of 466 lakes from the temperate zone to the arctic. Ecosystems 6:313-325

Jonasson, P.M. 1996. Limits for the life in the lake ecosystem. Verhandlungen der internationale Vereiningung fur Theoretische und Angewandte Limnologie 20:1-33

Kajak, Z. 1987. Determinants of maximum biomass of benthic Chironomidae (Diptera). Entomologica Scandinavica Supplements 29:303-308

Kanninen, A., J. Leka, V.M. Vallinkosk, and R. Ilvonen. 2003. Aerial photograph interpretation and field surveys of aquatic macrophytes in assessing the ecological status of small boreal lakes – preliminary results. TemaNord 547:123-126.

Kansanen, P.H., J. Aho, and L. Paasivirta, 1984. Testing the benthic lake type concept based on chironomid associations in some Finish lakes using multivariate statistical method. Annales Zoologici Fenici 21:55-76

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6:21-27

King, L., P. Barker, and R.I. Jones. 2000. Epilithic algal communities and their relationship to environmental variables in lakes of the English Lake District. Freshwater Biology 45:425-442

King, R.S., and C.J. Richardson. 2003. Integrating bioassessment and ecological risk assessment: An approach to developing numerical water-quality criteria. Environmental Management 31:795-809

Kirchner, W.R., and P.J. Dillon. 1975. An emperical method of estimating the retention of phosphorus in lakes. Water Resources Research 11:182-183

Kirk, J.T.O. 1994. Light and photosynthesis in aquatic ecosystems. Cambridge University Press, New York

Kornijow, R. 1997. The impact of predation by perch on the size-structure of Chironomus larvae-the role of vertical distribution of the prey in the bottom sediments, and habitat complexity. Hydrobiologia 342/343:207-213

Lang, C. 1998. Using oligochaetes to monitor the decrease of eutrophication: the 1982-1996 trend in Lake Geneva. Archiv Fur Hydrobiologie 141:447-458

Lang, C. 1999. Contrasting responses of oligochaetes (Annelida) and chironomids (Diptera) to the abatement of eutrophication in Lake Neuchatel. Aquatic Sciences 61:206-214

Lang, Y., J.M. Melackand J. Wang. 1981. Primary production and fish yield in Chinese ponds and lakes. Transactions of the American Fisheries Society 110:346-350

Larsen, D.P., and A.T. Herlihy. 1998. The dilemma of sampling streams for macroinvertebrate richness. Journal of the North American Benthological Society 17:359-366

Legendre, P., and L. Legendre. 1998. Numerical ecology. Elsevier, Amsterdam

Leka, J., K. Valta-Hulkkonen, A. Kanninen, S. Partanen, S. Hellsten, A. Ustinova, R. Ilvonen, and O. Airaksinen. 2003. Vesimakrofyytit järvien ekologisen tilan arvioinnissa ja seurannassa. Maastomenetelmien ja ilmakuvatulkinnan käyttökelpoisuuden arviointi Life Vuoksi-projektissa. Alueelliset ympäristöjulkaisut, pp. 312

Ligtvoet, W., and M.P. Grimm. 1992. Fish in clear water. Fish-stock development and management in Lake Volkerak/Zoom. How an estuary changed into freshwater lake: The water management of Lake Volkerak/Zoom. J.C. Hooghart, and C.W.S. Posthumus. Delft. Proceedings and Information, CHO-TNO, 46:69-84

Los, F.J., M. Ouboter, and R.J. Fokkink. 1997. Onderzoek naar de slibdynamiek van de Loosdrechtse Plassen met behulp van mathematische modellen. WL Delft Hydraulics, Delft

Lyche Solheim, A.L., T. Andersen, P. Brettum, T. Bækken, T. Bongard, F. Moy, T. Kroglund, F. Olsgard, B. Rygg, and E. Oug. 2004. BIOKLASS - Klassifisering av økologiske status i norske vannforekomster: Forslag til aktuelle kriterier og foreløpige grenseverdier mellom god og moderat økologisk status for utvalgte elementer og påvirkninger. Norsk institutt for vannforskning (NIVA), Oslo, Rapport LNR 4860-2004, pp. 67

Lyytikainen, T., and M. Jobling. 1998. The effect of temperature fluctuations on oxygen consumption and ammonia excretion of underyearling Lake Inari Arctic charr. Journal of Fish Biology 52:1186-1198

Mazumder, A., and K.E. Havens. 1998. Nutrient-chlorophyll-Secchi relationships under contrasting grazer communities of temperate versus subtropical lakes. Canadian Journal of Fisheries and Aquatic Sciences 55:1652-1662

McQueen, D.J., J.R. Post, and E.L. Miles. 1986. Trophic relationship in freshwater pelagic ecosystems. Canadian Journal of Fisheries and Aquatic Sciences 43:1571-1581

McQueen, D.J., M.R.S. Johannes, J.R. Post, T.J. Stewart, and D.R.S. Lean. 1989. Bottom-up and top-down impacts on freshwater pelagic community structure. Ecological Monographs 59:289-309

Moss, B., P. Johnes, and G. Phillips. 1996. The monitoring of ecological quality and classification of standing water in temperate regions: a review and proposal based on a worked scheme for British waters. Biological Review 71:301-339

Moss, B., D., Stephen, C. Alvarez, E. Becares, W. Van De Bund, S.E. Collings, E. Van Donk, E. De Eyto, T. Feldmann, C. Fernández-Aláez, M. Fernández-Aláez, R.J.M. Franken, F. García-Criado, E.M. Gross, M. Gyllström, L.A. Hansson, K. Irvine, A. Järvalt, J.P. Jensen, E. Jeppesen, T. Kairesalo, R. Kornijów, T. Krause, H. Künnap, A. Laas, E. Lill, B. Lorens, H. Luup, M.R. Miracle, P. Nõges, T. Nõges, M. Nykänen, I. Ott, W. Peczula, E.T.H.M. Peeters, G. Phillips, S. Romo, V. Russell, J. Salujõe, M. Scheffer, K. Siewertsen, H. Smal, C. Tesch, H. Timm, L. Tuvikene, I. Tonno, T. Virro, E. Vicente, and D. Wilson. 2003. The determination of ecological status in shallow lakes - a tested system (ECOFRAME) for implementation of the European Water Framework Directive. Aquatic Conservation: Marine and Freshwater Ecosystems 13:507-549

Muhammetoglu, A., and S. Soyupak. 2000. A three-dimensional water quality-macrophyte interaction model for shallow lakes. Ecological Modeling 133:161-180

Murphy, K.J. 2002. Plant communities and plant diversity in softwater lakes of northern Europe. Aquatic Botany 73:287-324

Murphy, K.J., and M.J.M. Hootsmans. 2002. Predictive modelling of aquatic community attributes: biomass, biodiversity, biointegrity and biomonitoring. Acta Limnologica Brasiliensia 14:43-60

Nabben, R.H.M., C.W.P.M. Blom and L.A.C.J. Voesenek. 1999. Resistance to complete submergence in Rumex species with different life histories: the influence of plant size and light. The New Phytologist 144:313-321

Navodaru, I., A.D. Buijse and M. Staras. 2002. Effects of Hydrology and Water Quality on the Fish Community in Danube Delta Lakes. International Review of Hydrobiology 87:329-348

Newman, P.J. 1998. Classification of surface water quality. Review of schemes used in EC member states. Heinemann Professional Publishing Ltd: Oxford

Nikolski, G.V. 1962. Ecologia pestilor, Bucuresti, pp. 362

Noges, P., L. Tuvikene, T. Feldmann, I. Tõnno, H. Künnap, H. Luup, J. Salujõe, and T. Nõges. 2003. The role of charophytes in increasing water transparency: a case study of two shallow lakes in Estonia. Hydrobiologia 506:567-573

OECD. 1982. Eutrophication of waters: monitoring, assessment and control. Organisation of Economic Co-operation and Development, Paris, pp. 154

Oosterberg, W., M. Sataras, A.D. Buijse, L. Bogdan, A. Constantinescu, H. Coops, J. Hanganu, B.W. Ibelings, G.A.M. Menting, I. Navodaru, L. Török. 2000. Ecological gradients in the Danube Delta lakes. Present state and man-induced changes. Lelystad, RIZA report 2000.015

Palmer, M.A., S.L. Bell and I. Butterfield. 1992. A botanical classification of standing waters in Britain: applications for conservation and monitoring. Aquatic Conservation: Marine and Freshwater Ecosystems 2:125-143

Phillips, G.L., D. Eminson, and B. Moss. 1978. A mechanism to account for macrophyte decline in progressively eutrophicated freshwaters. Aquatic Botany 4:103-126

Portielje, R., and D.T. Van der Molen. 1999. Relationships between eutrophication variables: from nutrient loading to transparency. Hydrobiologia 408:375-387

Premazzi, G., and G. Chiaudani. 1992. Ecological Quality of Surface Waters. Commission of the European Communities, pp. 124

Rasmussen, J.B., and J. Kalff. 1987. Empirical models for zoobenthic biomass in lakes. Canadian Journal of Fisheries and Aquatic Sciences 44:990-1001

Real, M. 2000. Chironomus species (Diptera, Chironomidae) in the profundal benthos of Spanish reservoirs and lakes: factors affecting distribution patterns. Freshwater Biology 43:1-18

Regier, H.A., R.L. Welcomme, and R.J. Steedman. 1998. Rehabilitation of degraded river ecosystems. Proceedings of the International Large River Symposium (LARS), Canadian Special Publication of Fisheries and Aquatic Sciences 106:86-97

Reynolds, C.S. 1984. The ecology of freshwater phytoplankton. Cambridge University Press, Cambridge

Reynolds, C.S. 1999. Modelling phytoplankton dynamics and its application to lake management. Hydrobiologia 396:123-131

Reynolds, C.S., V. Huszar, C. Kruk, L. Naselli-Flores, and S. Melo. 2002. Towards a functional classification of the freshwater phytoplankton. Journal of Plankton Research 24:417-428

Root, R.B. 1967. The niche exploitation pattern of the blue-gray gnatcatcher. Ecological Monographs 37:317-350

Rørslett, B. 1991. Principle determinants of aquatic macrophyte richness in northern european lakes. Aquatic Botany 39:173-193

Rosenberg, D.M. 1992. Freshwater biomonitoring and chironomidae. Netherlands Journal of Aquatic Ecology 26:101-122

Saether, O.A. 1979. Chironomid communities as water quality indicators. Holarctic Ecology, 2:65-74

Sand-Jensen, K., and M. Søndergaard. 1981. Phytoplankton and epiphyte development and their shading effect on submerged macrophytes in lakes of different nutrient status. Internationale Revue der gesamten Hydrobiologie 66: 529-552

Sas, H. 1989. Lake restoration by reduction of nutrient control. Academia Verlag Richartz, St. Augustin, pp. 497

Schaminee, J.H.J., E.J. Weeda and V. Westhoff. 1995. Die vegetatie van Nederland. Deel 2. Plantengemeeschapen van watersen moerassen en natte Leiden. Opulus Press, Uppsala, Leiden, pp. 357

Schaumberg, J., C. Scranz, G. Hofmann, D. Stelzer, and S. Schneider. 2004. Macrophytes and phytobenthos as indicators for ecological status in German lakes - contributions to the implementation of the WFD. Limnologica

Scheffer, M., S.H. Hosper, M.L. Meijer, and B. Moss. 1993. Alternative equilibria in shallow lakes. Trends in Ecology and Evolution 8:275-279

Scheffer, M. 1998. Ecology of Shallow Lakes. Chapman and Hall, London

Sculthorpe, C.D. 1967. The Biology of Aquatic Vascular Plants. Edward Arnold Editor, London

Sondergaard M, E. Jeppesen, J.P. Jensen, E. Bradshaw, H. Skovgaard, and S. Grunfeld. 2003. Vandrammedirektivet og danske soer Undertitel: Del 1: Sotyper, referencetilstand og okologiske kvalitetsklasser. Danmarks Miljøministeriet, Copenhagen

Spence, D.H.N., and J. Chrystal. 1970. Photosynthesis and zonation of freshwater macrophytes. I. Depth distribution and shade tolerance. New Phytologist 69:205-215

Swedish Environmental Protection Agency. 2000. Lakes and Watercourses: Environmental Quality Criteria. Report 5050, pp. 102

Takamura, K., and T. Iwakuma. 1990. Fate of macrozoobenthos in hypertrophic lakes: *in situ* rearing experiments with the larvae of Tokunagayusurika akamusi (Chironomidae). Archiv Fur Hydrobiologie, 119:65-78

Tilzer, M.M. 1988. Secchi Disk-Chlorophyll Relationships in a Lake with Highly Variable Phytoplankton Biomass. Hydrobiologia 162:163-171

US EPA. 1998. Lake and Reservoir Bioassessment and Biocriteria Technical Guidance Document (EPA 841-B-98-007). Office of Water, Washington DC

US-EPA. 1986. Ambient Water Quality Criteria for Dissolved Oxygen (Freshwater). EPA 440/5/86-003, U.S. Environmental Protection Agency, Washington, D.C.

US-EPA. 1997. Ambient Water Quality Criteria for Dissolved

Van de Bund, W., and D. Groenendij. 1994. Seasonal dynamics and burrowing of littoral larvae in relation to competition and predation. Archiv Fur Hydrobiologie, 132:213-225

Van de Bund, W., and E. Van Donk. 2002. Short-term and long-term effects of zooplanktivorous fish removal in a shallow lake: a synthesis of 15 years of data from Lake Zwemlust. Freshwater Biology 47:2380-2387

Van den Berg, M.S., H. Coops, M-L. Meijer, M. Scheffer, and J. Simons. 1997. Clear water associated with a dense Chara vegetation in the shallow and turbid lake Veluwemeer. In: E. Jeppesen, M. Søndergaard, M.Søndergaard, and K. Christoffersen (Eds.), The Structuring Role of Submerged Macrophytes in Lakes. Springer, New York, pp. 339-352

Van den Berg, M.S., M. Scheffer, E. Van Nes, and H. Coops. 1999. Dynamics and stability of Chara sp. and Potamogeton pectinatus in a shallow lake changing in eutrophication level. Hydrobiologia 409:335-342

Van den Berg, M.S., W. Joosse, and H. Coops. 2003. A statistical model predicting the occurrence and dynamics of submerged macrophytes in shallow lakes in the Netherlands. Hydrobiologia 506-509: 611-623

Van Donk, E., and W. Van de Bund. 2002. Impact of submerged macrophytes including charophytes on phyto- and zooplankton communities: allelopathy versus other mechanisms. Aquatic Botany 72: 261-274

Van Nes, E.K. 2002. Controlling complexity in individual-based models of aquatic vegetation and fish communities. Wageningen University, pp. 200

Verdonschot, P.F.M. 1996. Oligochaetes and eutrophication; an experiment over four years in outdoor mesocosms. Hydrobiologia 334:169-183

Weisner, S.E.B., J.A. Strand, and H. Sandsten. 1997. Mechanisms regulating abundance of submerged vegetation in shallow eutrophic lakes. Oecologia 109:592-599

Wiederholm, T. 1980. Using benthos for lake montioring. Journal of the Water Pollution Control Federation, 52:537-547

Willby, N.J., V.J. Abernethy, and B.O.L. Demars. 2000. Attribute-based classification of European hydrophytes and its relationship to habitat utilization. Freshwater Biology 43:43-74

Wunsam, S., and R. Schmidt. 1995. A diatom-phosphorus transfer function for Alpine and pre-alpine lakes. Memorie del Istituto Italiano di Idrobiologia 53:85-99